Climate change mitigation policy in Ecuador: Effects of land-use competition and transaction costs

Daniel V. Ortega-Pacheco[a,*], Andrew G. Keeler[b], Shiguo Jiang[c]

[a] ESPOL Polytechnic University, Escuela Superior Politécnica del Litoral, ESPOL, Facultad de Ciencias de la Vida, Campus Gustavo Galindo, Km. 30.5 Vía Perimetral, P.O. Box 09-01-5863, Guayaquil, Ecuador
[b] Department of Coastal Studies, East Carolina University, 850 NC 345, Wanchese, NC 27981, United States
[c] Department of Geography and Planning, University at Albany, State University of New York, Arts and Sciences 227, 1400 Washington Ave, Albany, NY 12222, United States

A B S T R A C T

This paper develops an econometric model to estimate the economic determinants of forest land conversion in Esmeraldas, Ecuador and evaluates the effect that transaction costs have on incentive-based climate change mitigation activities. Using a logistic share model, the results show a high extent of competition for the use of land between agricultural production for palm oil processing and land-based carbon mitigation through avoided emissions from deforestation. Our results show that the current policy of introducing conservation payments in Ecuador is not sufficient to significantly reduce deforestation, but that payment levels based on a plausible international carbon price would make an appreciable difference in the net sequestration contribution of forests in northern Ecuador. Knowledge generated from this research can inform site-specific land use policy design aimed at addressing the opportunity costs of tropical forest conservation as an important component of climate mitigation strategies.

1. Introduction

Reducing greenhouse gas (GHG) emissions—particularly from fossil fuel combustion—is the focus of domestic and international policies aimed at diminishing the risks of anthropogenic climate change. Emissions trading programs are likely to be the central incentive-based policy tool used to accomplish this task in developed nations. These programs are thought to have the capacity to bring about GHG reductions in a highly efficient way—but for ambitious targets they will still impose substantial economic costs. A number of studies have now suggested that land-based carbon sequestration credits can reduce the costs of meeting stringent GHG goals (e.g., Sohngen and Mendelsohn, 2003; Tavoni et al., 2007; Nabuurs et al., 2007; Kindermann et al., 2008). Many of the credits that these studies anticipate are derived from actions undertaken in developing countries.

Two critical concerns about the cost estimates in these studies have arisen. First, the individual studies described above assume that international credits can be generated without cost. However, it has been noted that transaction costs could raise the total cost of providing sequestration services through reductions in deforestation or forest degradation (see Antinori and Sathaye, 2007). A recent global analysis

indicated that most existing estimates of marginal costs of carbon sequestration are underestimated by up to 30 percent because transaction costs were not included (see Pearson et al., 2014). Second, many large-scale modeling studies estimating the cost of land-based credits have made critical assumptions about the elasticity of land supply in developing countries without having solid empirical evidence. Specifically, some studies (e.g., Sohngen and Mendelsohn, 2003) have used elasticity estimates from empirical studies in the United States, and the authors have transferred those to elasticity estimates to studies on other regions. As shown in Sohngen and Mendelsohn (2007), assumptions about land supply can have a great effect on the resulting supply of carbon credits available. Recently, Dang Phan et al. (2014) call for greater attention to transaction costs of policies to avoid deforestation in developing countries.

Improved estimates of how sequestration service provision responds to economic incentives that in real-world institutional settings is of high importance, given the centrality of these services to national and international efforts to efficiently reduce net GHG emissions. The high rate of deforestation in developing countries underscores the fact that understanding the financial constraints to conservation is essential in the near term (Fisher et al., 2011). Tavoni et al. (2007) suggest that

reductions in deforestation can reduce costs of meeting stringent carbon targets by 40% in coming years.

"REDD+" (Reducing Emissions from Deforestation and Forest Degradation or enhancement of carbon stocks) is a global mechanism under the United Nations Framework Convention on Climate Change that seeks to offset the opportunity costs of forest conservation through introducing economic incentives in the form of conditional payments (Phelps et al., 2010; Ghazoul et al., 2010). REDD+ will function through governments at the national or subnational level and could be implemented with either programmatic or project-based policies, depending on how land-use planning authority is distributed within each country (Busch et al., 2012).

In Ecuador, a payment for ecosystem services (PES) scheme called the Socio Bosque (Forest Partner) Program has been operating since 2009 across the country with the aim of introducing conservation incentives using farm-level contracts covering multiple environmental services, including carbon emissions from deforestation. Socio Bosque is a government program initially funded with fiscal resources. National authorities expect that global flows from REDD+ could contribute to finance the long term implementation of Socio Bosque.

Acknowledging the central importance of oil palm as a competing land use to forest in the developing world, researchers have explored the profitability of converting forest to oil palm versus conserving it for projects where incentives are introduced to reduce emissions from deforestation and forest degradation as an attempt to estimate smallholder opportunity costs to conservation (e.g., Butler et al., 2009; Venter et al., 2009; Persson and Azar 2010; Busch et al., 2015) and their spatial heterogeneity across the globe (Fisher et al., 2011). These estimates, though, do not fully account for transaction costs and, in the case of Tavoni et al. (2007) use the same land supply elasticity for different regions of the world. This paper uses an econometric model to estimate conversion of forest land into African Palm plantations in response to economic incentives in one developing country, Ecuador, and explicitly evaluates the effect that transaction costs have on land-based mitigation activities and resulting levels of GHG sequestration. Using this modeling approach, this research will explore how available policies might change future incentives and outcomes. Knowledge generated from this research aims at improving the understanding of the monetary gap between the costs of conserving forests and the joint benefits gained from converting them to African Palm plantations. Through sensitivity analysis, this paper models increasing payments for conservation and their effect on incentives to clear land and establish African Palm plantations.

Our paper studies African Palm in Ecuador for both substantive and practical reasons. Substantively, the focus of this paper in the province of Esmeraldas in northwestern Ecuador is based on: i) the significant institutional reforms implemented at the national level, ongoing capacity building and initiatives[1] for climate change mitigation including conservation payments programs similar to REDD+ (MMRREE 2007 and MAE 2010) and energy self-sufficiency through renewable energy, i.e. biofuels (Jull et al., 2007; Pelaez-Samaniego et al., 2007; República del Ecuador, 2004, 2007a, 2007b; Rothkopf, 2007), and ii) the large potential for palm oil production[2] given agroclimatic conditions, the relative amount of land available for conversion that could be linked to deforestation, and existing excess processing capacity of vegetable oil (ECLAC and GTZ 2004; Ludena et al., 2007; González, 2007;

PROECUADOR, 2016). Practically, this region contains a number of eco-zones, ranging from tropical lowland forests to dry highland forests, a fact which provide the variability needed to successfully estimate a land-use change model. The region also has good data sources for estimating a spatially explicit model. Moreover, this research could provide valuable inputs to development planning in a region marked by racial, ethnic and income inequalities and security concerns as well as different factors affecting distribution of opportunities and possibilities which ultimately shape the struggle for social and environmental justice in the context of climate vulnerability (see Rival, 2003; Johnson, 2007; Luque et al., 2013; Roa Ovalle, 2018 for a detail account).

Given the profitability of African Palm cultivation and processing, there has been concern in Ecuador and other parts of the world about its potential to drive deforestation and therefore have negative effects on GHG emissions, biodiversity, and social vulnerability (see Butler et al., 2009; Persson and Azar, 2010; Busch et al., 2015; Bateman et al., 2015; Vijay et al., 2016). Esmeraldas is the region with the biggest production of African palm in Ecuador (i.e., 1,711,585 MT representing 48.76% of the national production) and has the largest installed capacity for its processing industry (INEC, 2013; PROECUADOR, 2016). Ecuador is the worlds's fifth-largest exporter of palm oil, representing 1.8% of global exports. The increase in production has been the result of the expansion of the agricultural frontier driven by rising market prices rather than increased productivity. According to UNDP (2016), if this expansion continues, encroachment into primary forest will increase. In fact, Hazlewood (2012), critically documents the way that the expansion of palm plantations in this study area has displaced local practices that employ mixed uses of the forest (i.e. hunting, fishing, small-scale slash and mulch agriculture). A recent analysis shows that 27 different tax and financial incentives directly or indirectly affect palm oil production in the agriculture sector without considering the effects they have on deforestation, degradation or conservation of carbon stocks (UNEP, 2014).

Our methodology builds on earlier work by Plantinga et al. (1999) and Sohngen and Brown (2006). These efforts have developed land supply functions for the United States, using either county-level data or more specific plot-level data. Other studies have examined land use change in the Amazon basin (e.g., Nelson and Hellerstein, 1997; Pfaff, 1999), but these studies were conducted at a lower level of resolution than in our research.

Previous studies in Ecuador have explored land use change dynamics at different levels of specificity, determined land use drivers and patterns, and considered interdependent effects on food production, natural resources and provision of environmental services (e.g., de Koning et al., 1998; de Koning et al., 1999a; de Koning et al., 2003; de Koning et al., 1999b; Overmars et al., 2003; Curatola Fernández et al., 2015). The use of dynamic systems and agent-based modeling in the Amazon, Andes and Galápagos has added to this body of work by examining important pattern–process relations in complex ways (e.g., Mena et al., 2011; Curatola Fernández et al., 2015; Walsh and Mena 2016). Recent studies have attempted econometric modeling approaches for land-use change in dry forest of other regions of Ecuador that account for smallholder risk to explore the effect of conservation payments on portfolio diversification (e.g., Benitez et al., 2006; Knoke et al., 2015; Castro et al., 2015; Ochoa et al., 2016; Raes et al., 2016). These previous studies offer a useful point of departure for our modeling of land use change as a function of economic returns, and for the incorporation of transaction costs into the measure of those economic returns. Our results provide some evidence about the conditions under which policy alternatives and their associated transaction costs are effective in reducing GHG emissions in developing countries. This evidence is particularly relevant for situations where standing forest in relatively undeveloped areas faces clearing for livestock or agriculture activities and immediate returns to timber sales, a situation relevant to much of Amazonian South America and parts of Asia. By estimating changes in forestland use shares, we are able to make projections to

---

[1] Such as the National Strategy for Reducing Emissions from Deforestation and Forest Degradation and the ongoing national incentive-based program for forest conservation, Socio Bosque.

[2] By 2013, Ecuador represented the 0.9% of global oil production of palm oil encompassing 280,000 ha. Whereas in 2004, the production of Palm oil was 282,000 TM, in 2014 it was estimated in 484,000 TM (43% national consumption and 57% for export). Esmeraldas had the major development in sowed area of Palm oil.

assess the implications that shifts in land use shares may have on baseline carbon storage in the region. Finally, the land-use share and simulation model is used to examine the types of economic incentives that could be used to encourage the retention of forestland in the region.

2. Materials and methods

2.1. Calculation

We follow the central trend of the literature in that we model land use choice as being determined by relative rents, policy variables, and land characteristics (Miller and Plantinga, 1999). These factors have also been shown to be relevant to empirical studies exploring main determinants of land use change in Ecuador (see Sierra, 2001; Sierra and Stallings, 1998; Southgate et al., 2000; de Koning et al., 1998, 1999a,b, 2005). We use a land-use share model, following Hardie and Parks (1997) and Sohngen and Brown (2006), to examine agricultural and forestland in Esmeraldas province in northwestern Ecuador. Unlike most previous logistic models exploring the conversion of agricultural land to forestry in the United States (i.e., Hardie and Parks, 1997; Plantinga et al., 1999; Ahn et al., 2000), this paper focuses explicitly on naturally occurring forestland converted into agriculture use in a tropical developing country context. Given the relative importance of African Palm plantations as share of agricultural land in the region and the scale of its rents, we use it as a proxy to explore the opportunity cost of conservation.

We depart from Sohngen and Brown's (2006) multinomial logit model in that we do not disaggregate land use into types of forest uses. Sohngen and Brown's (2006) model analyzed different types of cultivated forest, whereas we propose to study agriculture land shares aggregated into one single management type (i.e. African palm plantations) and compare it to naturally occurring forest (i.e. primary and secondary forest). We are able to do this because the only other quantitatively significant land use – pasture – is seen only after land has been cleared and used for agriculture for some period of time (there is no direct conversion from forest to pasture) (see Wunder, 2000). Conversion to crops other than African palm, is collectively small enough that did not significantly affect conservation incentives (See Appendix C in Supplementary material). These other crops include cocoa, plantain, and tropical fruits. Our ability to restrict our choice to only two types of land uses is critical to our decision to employ a logistic model.

This study focuses on expected net returns, including those from government programs, that affect farmers' land use share response. Barr et al. (2011) reported land supply elasticities for the United States and Brazil as a way of assessing how much policy-induced increases—i.e. those in demand for biofuel, feedstocks, or agricultural CO_2 off-sets—will result in higher prices or expanded supply and, in turn, how those increases might affect deforestation rates. This study complements Barr et al. (2011), as it incorporates the effect of transaction costs on the effectiveness of government programs affecting land use choices.

Modeling forest rents for natural occurring tropical forest required a fundamentally different approach than that used for previous studies on land use change conversion in the United States. In those studies, returns to forest management were a function of long-term management of timber as a renewable asset (e.g. Sohngen and Brown, 2006). In our study site, however, there is no major sustainable plantation forestry: the rents for timber harvesting are associated with land clearing and conversion to agriculture in Ecuador (Wunder, 2000).[3] This means that

in our study, we expect higher forest rents to *increase* the likelihood of conversion to African palm plantations, whereas in the context of managed forestland, increases in the rents for forestry would be predicted as *decreasing* land conversion to African palm plantations.

We posit that there is some value in not converting the land, a notion that is related to the sum of the above factors; also, we propose that this value reflects positively on the significant amount of forestland in the initial period of our study. The variability in forestry and agricultural rents (as well as the other independent variable) explains differences in behavior across landowners and is used to predict changes in our model.

The proportion of land in each of these uses (forest and agriculture) in each parish is expressed as a binomial logistic function.[4] Consistent with previous efforts by Miller and Plantinga (1999), land use shares in the selected region can be estimated as a function of explanatory variables, including proxies for land rents from alternative uses, and institutional (i.e., tenure formalization, immigration rate and population density), and bio-physical factors (i.e., soil agricultural potential). As per Maddala (1983), the functional form for the binomial logit can be expressed as:

$$y = \frac{1}{1 + e^{-f(X)}}, \text{ i.e.,}$$
$$P_j = \frac{1}{1 + e^{-(\beta_0 + \beta_1 x_{1,i} + \beta_2 x_{2,i} + \cdots + \beta_k x_{k,i})}} \tag{1}$$

The left-hand side of Eq. (1) is the proportion of land allocated to usage j. X is the vector of independent variables and β is the vector of coefficients to be estimated. Under the assumption that P_j is distributed as a generalized extreme value distribution, the log-odds ratio (the ratio of P_j/P_m, for example) can be derived as a linear function of the parameters and expressed as follows:

$$\ln\left(\frac{P_j}{P_m}\right) = (\beta_j - \beta_m)X \tag{2}$$

As noted in Hardie and Parks (1997), and Plantinga et al. (1999), parameter estimates for β_j can be obtained by setting $\beta_m = 0$, assuming that the errors are normally and identically distributed. Thus, Eq. (1) can be transformed into a linear form with two different land uses (j and m), and expressed as follows:

$$L(P_j) = \ln\left(\frac{P_j}{1 - P_j}\right) = \beta_0 + \beta_1 x_{1,j} + \beta_2 x_{2,j} + \cdots + \beta_k x_{k,j}$$

$$\ln\left(\frac{P_j}{P_m}\right) = \beta X + \varepsilon \tag{3}$$

For the land uses proportion considered in our model, the specific equation estimated is:

$$\ln\left(\frac{\text{Forest}_i}{\text{Agriculture}_i}\right) = \beta_0 + \beta_1 x_{1i} + \beta_2 x_{2i} + \cdots + \varepsilon_i \tag{4}$$

where Forest_i: share of land in naturally occurring forest; Agriculture_i: share of agricultural land dedicated to African Palm cultivation only; x_{1i} x_{2i}...: independent explanatory variables (Table 1) indexed to parish i, including agriculture and forest extraction rents (which are combined into a single variable as ForAgRent), soil agricultural potential (Soil), population density (PopD), immigration rate (ImRate) and tenure (Tenure); $\beta = (\beta_0, \beta_1, ..., \beta_k)^T$ is a vector of unknown parameters to be estimated; ε: normally distributed, *i.i.d* error terms.

(*footnote continued*)

our interest in comparing only forest conversion to agriculture, we only consider the first year dedicated to wood extraction. The specification of our land share model will allow us to explore how the landowner would compare forest rents to agriculture rents.

[4] For ease of use, the share functions are specified in logistic form in order to restrict the estimated shares to the unit simplex.

---

[3] Note that this is different from Wunder's net present value analysis approach to estimated rental values for tropical forests during a fifteen-year period. He assumed that during a full forest conversion cycle, the first two years are dedicated to wood extraction (timber and charcoal), the next four to agriculture and the last ten to cattle-ranching. According to field observations and based on

Table 1
Variables used in the regression analysis.

Variable	Description
ForAgRent	The sum of values for forestland clearance value and returns to African palm
Soil	Soil agricultural potential
PopD	Population density
ImRate	Immigration rate
Tenure	Dummy variable representing parishes that have aggregated measures of land with and without formal tenure status

The independent variables included in the model were selected carefully. First, we constructed a pool of independent variables based on the literature (see for example, Plantinga et al., 1999; Lubowski et al., 2006; Sohngen and Brown, 2006) and on data availability. Second, we evaluated the log-linear specification using Ramsey's RESET test (Ramsey, 1969). Third, we conducted model selection by stepwise regression using criteria from the Akaike Information Criterion (Akaike, 1980).

Using the resulting parameter estimates, the proportion of land allocated to the two land uses can be predicted for each parish. The model can then be used to project future land use changes by altering the vector **X**. For instance, alternative future rental values resulting from the introduction of direct payments to incentivize forest conservation can be projected, and used to predict the relative area of land allocated to forest and agricultural land uses.

2.2. Data sources

Data for this model were drawn from several sources (Table 1). Information on the agricultural production units was obtained from the Ecuador Census of Agriculture (Censo Nacional Agropecuario, or CNA). These data were collected during the period 1999–2000, and provide information on several relevant attributes (including land use and land tenure) (INEC-MAG-SICA, 2003). The CNA survey data were originally collected at farm level and it is factored using the original CNA expansion factors. Data on the proportion of African Palm plantations and forest shares were drawn from land use maps developed by the Center for Integrated Inventory of Natural Resources (CLIRSEN, 2008). Data has been aggregated into Esmereldas's 62 parishes using appropriate weights (a full list of variables and data sources is given in Appendix A in Supplementary material).

Agricultural rents were estimated using information from African Palm budgets. We estimated production costs from regional budgets from the Ministry of Agriculture. The budget approach also allows us to include information about the mix of pesticides and fertilizers to calculate variable costs, which is information not available from the CNA. This is particularly important in the case of a commercial agricultural product like African Palm because costs associated with pesticides and fertilizers represent a significant share of total variable costs. We used annual average farm-gate prices (i.e., output prices) from Ministry of Agriculture data. While the CNA does not specifically include land values or rents, it did provide supplemental information on outputs from individual agricultural production units. Agricultural rents are estimated as present value of net returns per hectare of land for African Palm cultivation, using a 12% discount rate following common discount rates used in the literature (see Dang Phan et al., 2014).[5]

---

[5] In Ecuador, the National Secretary of Planning evaluates public investments in conservation programs and has fixed the rate on 12% to reflect the opportunity costs of public resources and ensure their most efficient allocation with an inherently selection bias towards project that generate greater economic benefits and lower costs in the long term. This rate has in fact been used to evaluate Ecuadorean forest project, and is line with the IPCC highest

As per Sierra (2001) who suggests that a large proportion of total harvests in Ecuador are locally consumed (> 50%), forest rents per hectare will be calculated as:

ForestRents$_i$ = Roundwood * P_i^s, i: 1… n where ForestRents$_i$ indicates rental value per hectare of forestland in parish i, Roundwood$_q$ is the average measure of roundwood per hectare of tropical forest in the region, and P_i^s is the stumpage price for parish i.

The stumpage price, P_i^s, can further be broken down as:

$$P^s = (P^l - C^h - C^m - C^t - C^k) * \eta$$

where, P^l is the end product price of the lumber, C^h are harvesting costs, C^m are milling costs, C^t are transportation costs, C^k are marketing costs, and η is the processing efficiency rate. Data for estimating forest rents per hectare have been drawn from Benítez (2005). This information has been supplemented by government statistics and data from peer-reviewed publications and interviews with local forestry experts.

Data on population density and immigration rates for each parish were obtained from the 2001[6] Population and Household Census of Ecuador (Censo de Población y Vivienda, or CPV) (INEC, 2001). Soil agricultural potential is extracted from the SIG-Agro (2008) agricultural potential map. The 18 types of agricultural potential are set to values ranging from 0 to 10 based on their texture, slope, and irrigation conditions. Land tenure status is captured through a dummy variable that takes on a value of 0 if the landowner does not have formal tenure. Any kind of official tenure status (i.e., land ownership with officially registered title, renter, tenant, communal or group ownership), is given a value of 1.[7] For a detailed description and construction of independent variables, please refer to Appendices A and B in Supplementary material.

3. Results

The results of the econometric analysis, based on 62 observations (one for each of Esmeraldas's parishes) are shown in Table 2. Many of the parameters are significant at the 1% or 5% level. As expected, higher rental values for forestland decrease the proportion of land devoted to the activity because they increase the returns to deforestation and land conversion. Higher African Palm plantation rents increase the proportion of land devoted to agriculture. Parishes with a higher soil agricultural potential have a lower proportion of forestland, as do parishes with a higher immigration rate.

Tenure status and population density have a positive effect on the proportion of forestland relative to African Palm plantation. The former is likely due to the fact that conversion to agriculture was a common strategy to ensure land use rights. Field observation confirm that access to enforceable tenure titles is perceived to reduce the incentive to convert forestland. The effect of population density may be related to patterns of localized population density and measures of land intensification associated with capital intensive productive systems like African Palm. The average proportion of land dedicated to forest in the region is estimated to be about 5.5 times the area relative to that dedicated to African Palm in the baseline period. In general, the model projects proportions of land use that are consistent with the observed patterns.

(*footnote continued*)
recommend discount rate 10–12% (see IPCC, 2007)

[6] Population and household census data were available for 1990, however, these socio-economic data for each parish were not interpolated using the available years to ensure that would it conform to the year of the CNA data. Given the proximity of collection data, we chose to use 2000 data and assumed that no major differences exist between the months when the data were collected for agricultural and population census was taken in year 2000.

[7] For simplicity's sake, it is assumed that land use decision, whether individually or communally, is equally affected by perceptions of rents. Further research should explore modeling different examples of land ownership and their relationship to observed land use shares.

Table 2
Parameter estimates of econometric land use model.

	Estimate[a]	Std. Error	t value	Pr(> |t|)		Mean	SD	Min	Max
(Intercept)	9.643022	2.669898	3.61	0.001	***				
ForAgRent	−0.0013358	0.0005613	−2.38	0.021	**	5,651.473	81,658.510	−284,327.90	118,319.20
Soil	−0.3294409	0.1009738	−3.26	0.002	***	3.597	1.325	0.65	6.64
PopD	0,0069783	0.0016839	4.14	0.000	***	28.628	53.646	0.39	412.34
ImRate	−1.353586	0.4267994	−3.17	0.002	**	0.421	0.397	0.00	1.69
Tenure	2.006865	1.028833	1.95	0.056	.	0.312	0.239	0.00	1.00

Significance: '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1.

[a] White's estimator is used to adjust the estimated coefficient standard errors using a variance–covariance matrix addressing heteroskedasticity (Zeileis, 2004).

3.1. Forest area and carbon projections

Projections of future land use are made by adjusting the rental rates for future time periods, and then re-projecting the area of land in alternative land uses. During 2000–2005, deforested land in Esmeraldas was mostly converted to African Palm cultivation (Ortega-Pacheco and Jiang, 2018). Based on field observations, forests in the region can be described as native primary or secondary forests that are rich in wood resources. Wood extraction is a highly labor-intensive activity. During the forest conversion cycle, the initial phase of land-use change (i.e., forest clearing) on a one-hectare plot over a one-year period is assumed to be dedicated to wood extraction. We assume that processing efficiency continues to improve, and that prices rise at relatively modest rates so that forestland rental rates increase at 1% per year.

By contrast, African Palm rents, which serve as our measure of agricultural rents, are assumed to increase at a higher rate. While agricultural yields for African Palm in this region have remained relatively stable according to the National Association of African Palm Growers and Processors (ANCUPA) 2005 Census, input prices have risen due to dollarization by more than 4–5% (ANCUPA-SIGAgro-MAG, 2005). Similarly, market prices have steadily risen about 1–2% per year. This is mainly due to increased world demand for vegetable oils, primarily in the food processing industry and biodiesel production (Gunstone, 2011). We believe that the evidence supports this continuing increase in the world palm oil price, and consequently we assume that African Palm rental values are likely to rise at a real rate of 2% per year over the next 30 years for our projections.

Table 3 presents the projected land areas for forest and the change predicted for the years 2010, and 2020, and 2030, relative to the baseline projected value in the year 2000. Because this study only considers carbon storage in forestland biomass and ignores carbon sequestration in agricultural soil, agricultural land areas and carbon storage on agricultural lands are not shown in Table 3. The results suggest that the current incentive system is likely to result in the conversion of an overwhelming majority of forestland to African Palm plantations in the next two decades. In fact, the results project that only 12.45% of forestland in 2000 will remain by the year 2030.

To calculate total emissions from forestland conversion, we follow Achard et al.'s (2002) approach. First, we consider existing regional figures of total carbon vegetation biomass derived from the actual biomass density without roots (i.e., 182 t per hectare of aboveground biomass for moist closed forest in Ecuador) (Brown, 1997). We add 20% for below-ground vegetation (root) biomass, accepting that root biomass varies considerably in tropical forests.[8] Second, carbon is assumed to be 50% of total biomass (Watson et al., 2000). The resulting computation indicates 109.2[9] tonnes of carbon (tC) per hectare[10] in existing uncleared forestland in Esmeraldas. Working with these assumptions, carbon stock projections for the baseline case are shown in Table 3. We assume that these values are uniform throughout Esmeraldas in uncleared forest; if there were significant variation and these could be effectively measured in implementing a carbon policy then our results would be affected. Future efforts should attempt to explore site specific measures of carbon for a spatial explicit analysis taking advantage of finer resolution land use modeling.

4. Discussion

Our results indicate that economic incentives driving land use change are consistent with the pattern of observed deforestation and consequent GHG releases in Esmeraldas. That is, as expected, higher forest rents increase the likelihood of conversion to African Palm agriculture. This reflects the underlying reality that measurable financial incentives in markets all work in the direction of converting forestland to vegetable oil production. The incentives to retain land as forest are not quantifiable for our study. They include the cash costs of clearing and planting African palm and the scarcity of capital and labor available for conversion in any given year, the prospect of greater returns from clearing in later periods, and unmeasured economic returns for uncleared forestland such as firewood, gathering renewable products, etc. We believe that these are similar enough across geography to not bias our results, but this lack of location-specific data is unfortunate and should be remedied in future research. Perhaps most important for the significant quantity of land held in communal ownership in Esmeraldas, the estimates do not include the non-market benefits of uncleared land, including ecosystem services (e.g., erosion protection, water purification) and existence and bequest values based on cultural norms and beliefs.

It is critical to test the sensitivity of these results, particularly for the assumptions underlying future scenarios. To assess the importance of land use price projections for the model's results, we explored a conservative scenario in which the African Palm price increases 1% annually, with no increase for forest clearing returns. The underlying assumption here and throughout the policy scenarios is that the percentage of increase in prices will result in a direct equal increase in total

Table 3
Forest area and carbon stocks.

	2000	2010	2020	2030	Average annual change
Forestland area (in million hectares)	0.861	0.615	0.303	0.107	0.025
Carbon stock in forests (in Mt CO_2)[a]	344,054	245,828	121,149	42,835	10,041

[a] Million tonnes carbon given; 1 Mt = 10^6 tonnes.

---

[8] The error range of such biomass estimates is thought to be as high as ± 30 to ± 60%.

[9] 109.2 = (182*1.2)/2.

[10] This result is consistent with Achard et al.'s (2002) regional estimates: 129 tonnes of carbon (tC) ha^{-1} 1 for the pan-Amazon and Central America region, 190 tC ha^{-1} for the Brazilian Amazon forests (23), 179 tC ha^{-1} for tropical moist Africa, and 151 tC ha^{-1} 1 for Southeast Asia.

rent for the respective land use. Under this conservative scenario and assumption regarding the elasticity of rents to price, about 62.96% of land initially in use as forest is projected to remain by 2030. This result points to a large difference in projected estimates relative to the 12.45% of land remaining in forest use, under what we believe is the more plausible scenario of a 2% annual increase in price for African Palm and 1% for forest price.

We now turn to an exploration of how available climate policies might change future economic incentives and outcomes. We model Socio Bosque payments as reductions in the net returns to clearing forest for timber. Increasing payments for conservation directly reduces the incentives to clear land and establish an African Palm plantation. The first policy design we examine is a single conditional payment covering multiple environmental services, following the Socio Bosque program. The payment per year depends on the area an individual landowner or community commits to conserve, with larger areas receiving a smaller per-hectare payment. The set of payments for conserving forestland examined is $30 per ha per year for 2000–2010, $40 per ha per year for 2010–2020, and $50 per ha per year for 2020–2030.[11] The payments are set to grow over time based on an expectation that i) national political support for payment for environmental service programs such a Socio Bosque will increase over time[12]; and ii) that Socio Bosque can be perceived by the international community as a performance based program that could effectively channeled through economic resources internationally mobilized by REDD +.

The second policy considered assumes same size conservation payments as the first policy, but it deducts transaction costs of program implementation, administration and activities across the 30-year analysis period. The variation in the relative shares of costs faced by the government agencies is partly related to the nature of the program, but also reflects the stage of its development (Dorward, 2001). Land-based mitigation policies require fixed cost implementation-type activities in their first years as the details of implementation are finalized and the program is set up; administration-related transactional activities are likely to increase in relative importance in later years. Here we posit that the balance will switch from "implementation" or "set-up" activities (such as promoting the program and entering into contracts) to more routine "maintenance" activities (like those related to checking compliance with contract terms). The average transaction costs for affecting programmatic approach is $5.53 per hectare per year. Transaction costs are set to diminish at a rate of 1 percent per year relative to the previous year. Transaction cost information[13] used in this study is based on Ortega-Pacheco and Keeler's (2018) empirical measurements for Esmeraldas.

The third policy we examine is changes in the size of payments. Here the structure of incentives simulates a REDD + payment. It assumes a potential value of a REDD + credit per ton of CO_2 avoided equal to the rental value of timber (i.e., opportunity cost of conservation). With the possible implementation of REDD + programs

enhancing the value of carbon sequestration that is linked to developed-country carbon markets, substantially higher payments are a plausible policy outcome. We model those by assuming an annual payment per hectare of $238 that is roughly equal to the net present value of carbon stock in forests reported by this research in existing uncleared forest-land in Esmeraldas based on average carbon price on the secondary market in 2010 (i.e., $1.32 per ton per year for emissions reductions generated under Kyoto Protocol mechanisms) discounted for 30 years (see World Bank, Ecofys and Vivid Economics, 2017). This a conservative hypothetical value for REDD + payments based on a price of carbon assumed constant relative to the general assumption used for sensitivity testing in project evaluations which assume price levels increase over time to reflect the increased marginal damage of GHG emissions in the future. Transaction costs of program implementation, administration and activities across a 30-year analysis period are the same as those used in the second scenario. To test for sensitivity of the value of payment, a variation of this policy is also examined in which only 50% of the opportunity cost of conservation is simulated as a REDD + payment.

The results in Table 4 show that conservation payments currently in place are low relative to the incentives for land conversion. In fact, any potential positive effect that could be associated with the introduction of conservation payments is outweighed by the likely trend of steady growth assumed for African Palm rental rates over time. This finding points to a need for conservation payment schemes in the study region that would, at the least, attempt to equal the "conservation opportunity costs"; otherwise, such schemes would not necessarily offer the landowner an economically competitive alternative to forest conversion. In fact, Table 3 suggests that about a range of 31–45% of land will continue as forest if REDD + payments are introduced and that 20–15% will remain as forest if conservative REDD + payments are introduced (50% of the total REDD + payment). The results of this policy scenario demonstrate that economic incentives based on a plausible international carbon price may represent a substantial difference in the incentive structure and would thus result in an appreciably impeded rate of forest conversion. Consequently, policy makers should attempt to design REDD + payments schemes valued at market reference price for avoided emissions, which may be significantly larger than the rents from timber.

Our results show that the effect of the transactions costs observed in Socio Bosque program in Esmeraldas are likely to have a fairly small negative impact on the incentive scheme. While efficient program design and minimization of transactions cost are important, they need not represent a critical impediment to implementing incentive-based conservation schemes. These results may be explained due to relative cost advantages arising from a programmatic approach to REDD + reported in this study rather than the project based consideration that has been mostly used in hypothetical scenarios where cost of development and maintenance have been estimated to be as high as $25 per ha (see Butler et al., 2009).

We can use the results of our modeling to provide a crude estimate of the payment it would take to remove all incentives to convert land to African palm and thus leave forested area unchanged for thirty years. The annual payment which would be predicted to halt land conversion − i.e. result in a value of zero for annual land clearing − is $1734 per ha per year. We caution that such a payment is a non-marginal change and does not account for adjustments that might take place in the price of palm oil and other key variables, and should not be regarded as a reliably exact prediction. It does, however, provide an indication that minimizing land conversion in Esmeraldas through conservation payments is likely to come at a very high price.

These results are in line with those of other studies examining the extent of competition between oil palm and conservation through REDD + projects in developing countries. Butler et al., 2009 estimated smallholder opportunity costs of REDD + in 17 sites in six tropical countries and found that REDD + will not be able to compete with

---

[11] Note that Socio Bosque was implemented in 2009. We are modeling a hypothetical situation where the specified payments were introduced in 2000, in order to examine how resulting deforestation rates would have differed over that past decade and how they would be predicted to change in the future.

[12] One limitation of this analysis is that it relies on non-marginal projections and that the elasticity of conversion would be likely to drop as the share of land devoted to African palm cultivation increases. In addition, the value of ecosystem services provided by a hectare of forestland should be expected to rise as the amount of forested land drops.

[13] Our study area includes both individual and communal private tenure structures. Although a large proportion of land is privately owned, it is important to acknowledge that the effect of transaction costs on rents can be affected. This paper is conservative in its modeling, as it transfers transaction costs from private landowners, which is perceived to be higher per unit per hectare, as compared to communal lands.

Table 4
Changes in forest area and carbon stock between 2000 and 2030 under alternative scenarios.

	Baseline	Without conservation payments	Conservation payments only	Conservation payments and transaction costs	REDD+ payments and transaction costs	REDD+ (half value-conservative) payments and transaction costs
Forestland area (in thousand hectares)	860,840	107,175	120,263	119,072	270,755	173,418
			(12.21%)[a]	(11.10%)		
Carbon stock in forests (in Mega ton CO_2)	344,054	42,835	48,066	47,590	108,213	69,310
Avoided emissions (in Mega ton CO_2)			5231	4755	60,147	21,721

[a] Quantities between parentheses indicate percentage change due to the policy (or lack thereof). There is also an approximate 1% reduction in forest area due to consideration of policy transaction costs.

African palm agriculture. Fisher et al. (2011) show that the profitability of logging, in combination with potential profits from subsequent conversion to palm-oil production far exceeds foreseeable revenues from a global carbon market and other ecosystem-service payment mechanisms. These authors argue that previous studies have underestimated the gap between conservation costs and conversion benefits in Southeast Asia and showed that carbon payments through a REDD+ mechanism or carbon market would have to offer between $22–$28 per tCO_2 to overcome the profit gained from logging; $46–$48 to overcome the opportunity cost of conservation with respect to both timber and oil palm and $40–$42 per tCO_2 including setup and monitoring costs. These ranges represent one-time payments to outweigh net returns from oil palm over a 25-year life cycle whereas our research explores annual payments across a 30-year analysis period. If annualized, Fisher et al. (2011) carbon payments would not necessarily be that much higher than the conservative carbon price used here (i.e., $1.32 per ton per year). The difference may be explained by the fact that opportunity costs in Malaysia seems higher even though calculations of the potential returns from oil-palm plantations in this region vary widely, from $4000 per hectare to $29,000 per hectare under a range of assumptions.

One limitation of this analysis is that our model does not account for landholder risk due to prices and yield variability. Therefore, future research in Esmeraldas should develop econometric estimates of risk following previous research in other regions of Ecuador to better understand its effect on land use decisions (e.g., Benitez et al., 2006; Knoke et al., 2015; Castro et al., 2015; Ochoa et al., 2016; Raes et al., 2016). Such studies have found that the required payment per hectare to guarantee that sustainable land uses dominates all other alternatives ranges from $55 to $89. Likewise, future analyses should rely on multiperiod data (see Lubowski et al., 2006) to analyze whether econometric modeling outperforms extrapolation of historical trends in predicting future values and the effect of diversification and transaction costs at the landscape level. Due to the high interest in conservation in Ecuador, future research should also provide insights into how of alternative approaches can be used effectively under given sets of circumstances to understand the aggregated impact of carbon uptake and conservation policies (see van Kooten and Sohngen, 2007). Lastly, we assume that tenure data and transportation infrastructure affecting rents measure can capture wealth differentials influencing land use decisions. We acknowledge limits that this assumption may pose in the generalizability of our results given the complex relationships governing wealth, poverty and inequalities in our study region. Future research should explore modeling approaches that incorporate different measures of wealth differential driving land use choices to effectively inform carbon policy design.

5. Conclusion

Land-based climate change mitigation actions undertaken in developing countries could generate carbon credits that can would reduce the costs of meeting GHG-reduction goals. This paper reports empirical measures that address two critical concerns about the potential that land-based credits may have for climate policy. First, a logistic share model is used to produce a model predicting the share of land in forestland and agricultural land in Northwestern Ecuador. Second, using the estimated model, this paper evaluates the effect that different levels of rents have on the conversion of standing forest to African Palm agriculture. It also estimates the impact that conservation payments and associated transactions costs are likely to have on land-based mitigation activities. In general, results show a high extent of competition with respect to the use of land between mitigation options. The net effect of transaction costs is minimal, and may not represent a critical impediment to reducing the scale of incentives to stop land conversion into African Palm agriculture in this setting. However, our results reaffirm that a fuller and more nuanced understanding of the magnitude of financial conservation incentives that affect behavior is critical if the international community is to more fully understand the relationship between payment levels and outcomes in REDD+.

Using this paper's empirical model parameter estimates, projections and sensitivity and policy analysis, we demonstrate that emissions from forest sinks can be reduced if conservation payments are introduced. However, the results show that significant payments would be required to make a substantial difference in the magnitude of deforestation for agriculture.

Finally, these results raise an interesting issue regarding the storage of carbon on the landscape and the extent of competition and complementarities, in light of crediting schemes for net emissions from forest-to-agricultural conversion that is associated with biofuels crops. Kyoto Protocol rules only considered credits for additional storage of carbon on the landscape, without considering emissions avoidance from forest or net avoided emissions from land conversion. Our sensitivity analysis on elasticity of land supply suggests that the results are highly sensitive to the estimated rental rates, as well as to the underlying assumptions concerning the rate of price changes over time. These results illustrate the trade-offs that could arise when designing policies to enhance terrestrial sequestration in light of the Paris Agreement. If it is just avoided emissions that are to be credited, or if net emissions from forest to agricultural conversion are credited, then economic incentives for establishment of agricultural land for biofuels processing can be a useful tool for enhancing carbon storage and emissions avoidance. However, if credits are also provided for emission offsets in the energy or transportation sectors (e.g. fuel vs. biofuel substitution), the analysis suggests that in the short term there would be incentives to expand the stock of African Palm and, thus, to decrease forestland. It is beyond the scope of this paper to conduct a full life-cycle analysis of energy uses during harvesting, transportation, and the processing of biodiesel products, however, additional research is needed to assess the full net mitigation potential of these alternative scenarios.

Acknowledgments

This research was conducted with the support of Climate, Water and Carbon Program, the Mershon Center for International Security Studies, and the Robert W. Backoff Research Award of The Ohio State University. Field research was implemented with the economic support of the in Latin America and the Caribbean Environmental Economics Program (LACEEP). LACEEP provided additional support in study design and the collection, analysis, and interpretation of data. The preparation of this article has been developed with the support of ESPOL and State University of New York at Albany. Funding sources had no involvement in the decision to submit the article for publication. Lastly, we thank Allan Blackman, Juan Robalino and other researchers comments and suggestions to preliminary versions of this document.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.landusepol.2018.10.015.

References

Achard, F., Eva, H.D., Stibig, H.-J., Mayaux, P., Gallego, J., Richards, T., Malingreau, J., 2002. Determination of deforestation rates of the world's humid tropical forests. Science 297 (5583), 999–1002.

ANCUPA-SIGAgro-MAG, 2005. Censo de Palmicultores. Quito, Ecuador: Asociación Nacional de Cultivadores de Palma Africana (ANCUPA) y Ministerio de Agricultura, Ganadería, Acuacultura y Pesca (MAG), Unidad de Sistemas de Información Geográfica (SIG-Agro).

Ahn, S., Plantinga, A.J., Alig, R.J., 2000. Predicting future forestland area: a comparison of econometric approaches. For. Sci. 46 (3), 363–376.

Akaike, H., et al., 1980. Likelihood and the Bayes procedure. In: Bernardo, J.M. (Ed.), Bayesian Statistics. University Press, Valencia, pp. 143–166.

Antinori, C., Sathaye, J., 2007. Assessing Transaction Costs of Project-based Greenhouse Gas Emissions Trading. Lawrence Berkeley National Laboratory, Berkeley, CA LBNL-57315.

Barr, K., Babcock, B., Carriquiry, M., Nasser, A., Harfuch, L., 2011. Agricultural land elasticities in the United States and Brazil. Appl. Econ. Perspect. Policy 33 (3), 449–462.

Bateman, I.J., Coombes, E., Fitzherbert, E., Binner, A., Bad'ura, T., Carbone, C., Fisher, B., Naidoo, R., Watkinson, A.R., 2015. Conserving tropical biodiversity via market forces and spatial targeting. Proc. Natl. Acad. Sci. U.S.A. 112 (24), 7408–7413.

Benítez, P., 2005. Cost-benefit analysis of forestry-based carbon sequestration in northwest Ecuador. Essays on the Economics of Forestry-Based Carbon Mitigation. PhD Dissertation. Wageningen University, Wageningen, The Netherlands.

Benitez, P.C., Kuosmanen, T., Olschewski, R., van Kooten, G.C., 2006. Conservation payments under risk: a stochastic dominance approach. Am. J. Agric. Econ. 88 (1), 1–15.

Brown, S., 1997. Estimating Biomass and Biomass Change of Tropical Forests: A Primer. FAO Forest Paper 134.

Busch, J., Lubowski, R.N., Godoy, F., Steininger, M., Yusuf, A.A., Austin, K., Hewson, J., Farid, M., Boltz, F., 2012. Structuring economic incentives to reduce emissions from deforestation within Indonesia. Proc. Natl. Acad. Sci. U.S.A. 109 (4), 1062–1067.

Busch, J., Ferretti-Gallon, K., Engelmann, J., Wright, M., Austin, K.G., Stolle, F., Turubanova, S., Potapov, P.V., Margono, B., Hansen, M., Baccini, A., 2015. Reductions in emissions from deforestation from Indonesia's moratorium on new oil palm, timber, and logging concessions. Proc. Natl. Acad. Sci. U.S.A. 112 (5), 1328–1333.

Butler, R.A., Koh, L.P., Ghazoul, J., 2009. REDD in the red: palm oil could undermine carbon payment schemes. Conservation Letters 2 (2), 67–73.

CLIRSEN, 2008. Mapas de uso y cobertura vegetal, forestal y deforestación a nivel nacional año 2000. Centro de levantamientos integrados de recursos naturales por sensores remotos (CLIRSEN).

Castro, L.M., Calvas, B., Knoke, T., Habibullah, M., Mosandl, R., Medina, C., 2015. Ecuadorian banana farms should consider organic banana with low price risks in their land-use portfolios. PLoS One 10 (3).

Curatola Fernández, G., Obermeier, W., Gerique, A., Sandoval, M., Lehnert, L., Thies, B., Bendix, J., 2015. Land cover change in the Andes of southern Ecuador—patterns and drivers. Remote Sens. 7 (3), 2509–2542.

de Koning, G.H.J., Veldkamp, A., Fresco, L.O., 1998. Land use in Ecuador: a statistical analysis at different aggregation levels. Agric. Ecosyst. Environ. 70 (2–3), 231–247.

de Koning, G.H.J., Veldkamp, A., Fresco, L.O., 1999a. Exploring changes in Ecuadorian land use for food production and their effects on natural resources. J. Environ. Manage. 57 (4), 221–237.

de Koning, G.H.J., Verburg, P.H., Veldkamp, A., Fresco, L.O., 1999b. Multi-scale of land use change dynamics in Ecuador. Agric. Syst. 61 (2), 77–93.

de Koning, G.H.J., Veldkamp, A., López-Ulloa, M., 2003. Quantification of carbon sequestration in soils following pasture to forest conversion in northwestern Ecuador. Glob. Biogeochem. Cycles 17 (4), 1098.

de Koning, F., Olschewski, R., Veldkamp, E., Benítez, P., López-Ulloa, M., Schlichter, T., de Urquiza, M., 2005. The ecological and economic potential of carbon sequestration in forests: examples from South America. Ambio 34 (3), 224–229.

Dang Phan, T.-H., Brouwer, R., Davidson, M., 2014. The economic costs of avoided deforestation in the developing world: a meta-analysis. J. For. Econ. 20, 1–16.

Dorward, A., 2001. The effects of transaction costs, power and risk on contractual arrangements: a conceptual framework for quantitative analysis. J. Agric. Econ. 52 (2), 59–73.

ECLAC and GTZ, 2004. Renewable Energy Sources in Latin America and the Caribbean: Situation and Policy Proposals. Economic Commission for Latin American and the Caribbean (ECLAC) and German Agency for Technical Cooperation (GTZ), Santiago.

Fisher, B., Edwards, D.P., Giam, X., Wilcove, D.S., 2011. The high costs of conserving Southeast Asia's lowland rainforests. Front. Ecol. Environ. 9 (6), 329–334.

Ghazoul, J., Butler, R.A., Mateo-Vega, J., Koh, L.P., 2010. REDD: a reckoning of environment and development implications. Trends Ecol. Evol. 25 (7), 396–402.

González, M., 2007. Programa de Biocombustibles en el Ecuador – Ministerio de Minas y Petróleos & Ministerio de Agricultura y Ganadería. In Biofuels as alternative energy: a regional outlook, edited by ICTSD, MICIP, UNEP y Ministerio del Ambiente. Quito, Ecuador: ICTSD, CEDA, IIED, FLACSO and CAN.

Gunstone, F., 2011. Production and Trade of Vegetable Oils. In: Gunstone Frank, D. (Ed.), Vegetable Oils in Food Technology: Composition, Properties and Uses, 2nd ed. Blackwell Publishing Ltd., 1-24.

Hardie, I.W., Parks, P.J., 1997. Land use with heterogeneous land quality: an application of an area base model. Am. J. Agric. Econ. 79 (2), 299–310.

Hazlewood, J.A., 2012. CO_2 lonialism and the "unintended consequences" of commoditizing climate change: geographies of hope amid a sea of oil palms in the northwest ecuadorian pacific region. J. Sustain. For. 31 (1–2), 120–153.

INEC, 2001. Censo de Población y Vivienda 2001. Retrieved from http://www.ecuadorencifras.gob.ec/base-de-datos-censo-de-poblacion-y-vivienda-2001/.

INEC, 2013. Encuesta de Superficie y Producción Agropecuaria Continua 2013. Retrieved from http://www.ecuadorencifras.gob.ec/documentos/web-inec/Estadisticas_agropecuarias/espac/espac%202013/InformeejecutivoESPAC2013.pdf.

INEC-MAG-SICA, 2003. III Censo Nacional Agropecuario. (CD-Rom). Instituto Nacional de Estadística y Censos (INEC), Ministerio de Agricultura, Ganadería, Acuacultura y Pesca, Proyecto Sistema de Información y Censo Agropecuario (MAG-SICA).

IPCC, 2007. Climate Change 2007: Working Group III: Mitigation of Climate Change. Retrieved from https://www.ipcc.ch/publications_and_data/ar4/wg3/en/ch2s2-4-2-1.html.

Johnson, E., 2007. Schooling, blackness and national identity in Esmeraldas, Ecuador. Race Ethnicity Educ. 10 (1), 47–70.

Jull, C., Redondo, P.C., Mosoti, V., Vapnek, J., 2007. Recent trends in the law and policy of bioenergy production, promotion and use. Legislative Study 95, Food and Agriculture Organization of the United Nations. FAO, Rome.

Kindermann, G., Obersteiner, M., Sohngen, B., Sathaye, J., Andrasko, K., Rametsteiner, E., Schlamadinger, B., Wunder, S., Beach, R., 2008. Global cost estimates of reducing carbon emissions through avoided deforestation. Proc. Natl. Acad. Sci. 105 (30), 10302–10307.

Knoke, T., Paul, C., Härtl, F., Castro, L.M., Calvas, B., Hildebrandt, P., 2015. Optimizing agricultural land-use portfolios with scarce data—a non-stochastic model. Ecol. Econ. 120, 250–259.

Lubowski, R.N., Plantinga, A.J., Stavins, R.N., 2006. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. J. Environ. Econ. Manage. 51, 135–152.

Ludena, C., Razo, C., Saucedo, A., 2007. Biofuels Potential in Latin America and the Caribbean: Quantitative Considerations and Policy Implications for the Agricultural Sector. In: Paper prepared for the American Association of Agricultural Economics Annual Meeting. July 29–August 1, Portland, OR.

Luque, A., Edwards, G.A.S., Lalande, C., 2013. The local governance of climate change: new tools to respond to old limitations in Esmeraldas, Ecuador. Local Environ. 18 (6), 738–751.

Maddala, G.S., 1983. Limited-Dependent and Qualitative Variables in Econometrics. Cambridge University Press, Cambridge.

Mena, C.F., Walsh, S.J., Frizzelle, B.G., Xiaozheng, Y., Malanson, G.P., 2011. Land use change on household farms in the Ecuadorian Amazon: design and implementation of an agent-based model. Appl. Geogr. 31 (1), 210–222.

Miller, D.J., Plantinga, A.J., 1999. Modeling land use decisions with aggregate data. Am. J. Agric. Econ. 81 (1), 180–194.

Nabuurs, G.J., O. Masera, K. Andrasko, P. Benitez-Ponce, R. Boer, M. Dutschke, E. Elsiddig, J. Ford-Robertson, P. Frumhoff, T. Karjalainen, O. Krankina, W.A. Kurz, M. Matsumoto, W. Oyhantcabal, N.H. Ravindranath, M.J. Sanz Sanchez, X. Zhang, 2007: Forestry. In Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [B. Metz, O.R. Davidson, P.R. Bosch, R. Dave, and L.A. Meyer (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Nelson, G.C., Hellerstein, D., 1997. Do roads cause deforestation? Using satellite images in econometric analysis of land use. Am. J. Agric. Econ. 79 (1), 80–88.

Ochoa, M., WS, Paul, C., Castro, L.M., Valle, L., Knoke, T., 2016. Banning goats could exacerbate deforestation of the Ecuadorian dry forest – how the effectiveness of conservation payments is influenced by productive use options. Erdkunde 70 (1), 49–67.

Ortega-Pacheco, D., Jiang, S., 2018. Climate policy: spatial explicit heterogeneity matters – the case of tropical deforestation at Northwestern Ecuador. Manuscript submitted for publication.

Ortega-Pacheco, D., Keeler, A., 2018. Transaction costs and carbon sequestration: measurement and policy. Manuscript submitted for publication.

Overmars, K.P., de Koning, G.H.J., Veldkamp, A., 2003. Spatial autocorrelation in multi-

scale land use models. Ecol. Model. 164 (2–3), 257–270.

PROECUADOR, 2016. Sector analysis: palm oil and products. Instituto de Promoción de Exportaciones e Inversiones (PROECUADOR). Quito, 18 p.

Pearson, T.R.H., Brown, S., Sohngen, B., Henman, J., Ohrel, S., 2014. Transaction costs for carbon sequestration projects in the tropical forest sector. Mitigation Adapt. Strategies Glob. Change 19 (8), 1209–1222.

Pelaez-Samaniego, M., Garcia-Perez, M., Cortez, L., Oscullo, J., Olmedo, G., 2007. Energy sector in Ecuador: Current status. Energy Policy 35 (8), 4177–4189.

Persson, U.M., Azar, C., 2010. Preserving the world's tropical forests—a price on carbon may not do. Environ. Sci. Technol. 44 (1), 210–215.

Pfaff, A., 1999. What drives deforestation in the Brazilian Amazon? Evidence from satellite and socio-economic data. J. Environ. Econ. Manage. 37, 26–43.

Phelps, J., Webb, E.L., Agrawal, A., 2010. Does REDD+ threaten to recentralize forest governance? Science 328 (5976), 312–313.

Plantinga, A.J., Mauldin, T., Miller, D.J., 1999. An econometric analysis of the costs of sequestering carbon in forests. Am. J. Agric. Econ. 81 (4), 812–824.

Raes, L., D'Haese, M., Aguirre, N., Knoke, T., 2016. A portfolio analysis of incentive programmes for conservation, restoration and timber plantations in Southern Ecuador. Land Use Policy 51, 244–259.

Ramsey, J.B., 1969. Tests for specification errors in classical linear least-squares regression analysis. J. R. Stat. Soc. Ser. B (Methodol.) 31 (2), 350–371.

República del Ecuador, 2004. Decreto Ejecutivo No. 2332: R.O. No. 482. 15 de diciembre de 2004.

República del Ecuador, 2007a. Decreto Ejecutivo No. 146: R.O. No. 39, 12 de marzo de 2007.

República del Ecuador, 2007b. Ley de Fomento de los Biocombustibles.: Congreso Nacional del Ecuador, Ley No. 28-148. 14 de Noviembre de 2007.

Rival, L., 2003. The meanings of forest governance in Esmeraldas, Ecuador. Oxford Dev. Stud. 31 (4), 479–501.

Roa Ovalle, I.E., 2018. La reconfiguración de las luchas afro en San Lorenzo (Ecuador) frente a la expansión de la palma aceitera y la transnacionalización del conflicto colombiano. Polisemia 13 (23), 49–62.

Rothkopf, G., 2007. Global Landscape - South America. A Blueprint for Green Energy in the Americas. Inter-American Development Bank, Washigton, DC.

SIG-Agro, 2008. Información cartográfica temática año 2000. (CD-Rom). Ministerio de Agricultura, Ganadería, Acuacultura y Pesca, Unidad de Sistemas de Información Geográfica (SIG-Agro).

Sierra, R., Stallings, J., 1998. The dynamics and social organization of tropical deforestation in Northwest Ecuador, 1983–1995. Hum. Ecol. 26 (1), 135–161.

Sierra, R., 2001. The role of domestic timber markets in tropical deforestation and forest degradation in Ecuador: implications for conservation planning and policy. Ecol.

Econ. 36 (2), 327–340.

Sohngen, B., Brown, S., 2006. The influence of conversion of forest types on carbon sequestration and other ecosystem services in the south central United States. Ecol. Econ. 57, 698–708.

Sohngen, B., Mendelsohn, R., 2003. An optimal control model of forest carbon sequestration. Am. J. Agric. Econ. 85 (2), 448–457.

Sohngen, B.R. Mendelsohn, 2007. A Sensitivity Analysis of Carbon Sequestration. Chapter 19 in Human-Induced Climate Change: An Interdisciplinary Assessment. Edited by M. Schlesinger, H.S., Kheshgi, J., Smith, F.C. de la Chesnaye, J.M., Reilly, T. Wilson, and C. Kolstad. Cambridge: Cambridge University Press.

Southgate, D., Salazar-Canelos, P., Camacho-Saa, C., Stewart, R., 2000. Markets, institutions and forestry: the consequences of timber trade liberalisation in Ecuador. World Development 28 (11), 2005–2012.

Tavoni, M., Sohngen, B., Bosetti, V., 2007. Forestry and the carbon market response to stabilize climate. Energy Policy 35 (11), 5346–5353.

UNDP, 2016. Priming Financial and Land-Use Planning Instruments to Reduce Emissions from Deforestation. Proposal approved by the Green Climate Fund – FP019: Decision B.14/17. 80 p. Retrieved from: https://www.greenclimate.fund/documents/20182/574760/Funding_proposal_-_FP019_-_UNDP_-_Ecuador.pdf/e586b720-abc1-41e2-ac9d-dc6047ce77c7.

UNEP, 2014. Fiscal policy in the Palm Oil Sector and its Implications for REDD+. UNEP Finance Initiative and Ministry of Environment of Ecuador. Ecuador, Quito.

van Kooten, G.C., Sohngen, B., 2007. Economics of forest ecosystem carbon sinks: a review. Int. Rev. Environ. Resour. Econ. 1 (3), 237–269.

Venter, O., Meijaard, E., Possingham, H., Dennis, R., Sheil, D., Wich, S., Hovani, L., Wilson, K., 2009. Carbon payments as a safeguard for threatened tropical mammals. Conserv. Lett. 2 (3), 123–129.

Vijay, V., Pimm, S.L., Jenkins, C.N., Smith, S.J., 2016. The impacts of oil palm on recent deforestation and biodiversity loss. PLoS One 11 (7).

Walsh, S.J., Mena, C.F., 2016. Interactions of social, terrestrial, and marine sub-systems in the Galapagos Islands, Ecuador. Proc. Natl. Acad. Sci. U.S.A. 113 (51), 14536–14543.

Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J., Dokken, D.J. (Eds.), 2000. Land Use, Land-Use Change, and Forestry. Intergovernmental Panel on Climate Change (IPCC), Special report. Cambridge University Press, UK 375 pp.

World Bank, Ecofys and Vivid Economics, 2017. State and Trends of Carbon Pricing 2017. World Bank, Washington, DC. Retrieved from: https://openknowledge.worldbank.org/bitstream/handle/10986/28510/wb_report_171027.pdf?sequence=7&isAllowed=y.

Wunder, S., 2000. The Economics of Deforestation. Palgrave Macmillan UK, London.

Zeileis, A., 2004. Econometric Computing with HC and HAC Covariance Matrix Estimators. Journal of Statistical Software 11, 1–17.